Burn me twice, shame on who? Interactions between successive forest fires across a temperate mountain region

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Abstract. Increasing rates of natural disturbances under a warming climate raise important questions about how multiple disturbances interact. Escalating wildfire activity in recent decades has resulted in some forests re-burning in short succession, but how the severity of one wildfire affects that of a subsequent wildfire is not fully understood. We used a field-validated, satellite-derived, burn-severity atlas to assess interactions between successive wildfires across the US Northern Rocky Mountains a 300,000-km^2 region dominated by fire-prone forests. In areas that experienced two wildfires between 1984 and 2010, we asked: (1) How do overall frequency distributions of burn-severity classes compare between first and second fires? (2) In a given location, how does burn severity of the second fire relate to that of the first? (3) Do interactions between successive fires vary by forest zone or the interval between fires? (4) What factors increase the probability of burning twice as stand-replacing fire? Within the study area, 138,061 ha burned twice between 1984 and 2010. Overall, frequency distributions of burn severity classes (low, moderate, high; quantified using relativized remote sensing indices) were similar between the first and second fires; however burn severity was 5–13% lower in second fires on average. Negative interactions between fires were most pronounced in lower-elevation forests and woodlands, when fire intervals were <10 yr, and when burn severity was low in the first fire. When the first fire burned as high severity and fire intervals exceeded 10–12 yr, burn-severity interactions switched from negative to positive, with high-severity fire begetting subsequent high-severity fire. Locations most likely to experience successive stand-replacing fires were high-elevation forests, which are adapted to high-severity fire, and areas conducive to abundant post-fire tree regeneration. Broadly similar severities among short-interval "re-burns" and other wildfires indicate that positive severity feedbacks, an oft-posited agent of ecosystem decline or state shift, are not an inevitable outcome of re-burning. Nonetheless, context-dependent shifts in both the magnitude and direction of wildfire interactions (associated with forest zone, initial burn-severity, and disturbance interval) illustrate complexities in disturbance interactions and can inform management and predictions of future system dynamics.

Key words: burn severity; conifer forests; disturbance interactions; feedbacks; linked disturbances; re-burn; wildfire.

Introduction

Natural disturbances (e.g., fires, insect outbreaks) are integral to the structure and function of many forests worldwide (Attiwill 1994), but climate warming may increase the frequency, size, and severity of disturbances beyond the response capacity of forest ecosystems (Millar and Stephenson 2015). Disturbances that are primarily climate-driven, such as forest fires, are expected to increase in many regions globally over the next century (Moritz et al. 2012), especially in conifer forests of western North America (Westerling et al. 2011, Barbero et al. 2015). Such increases will inevitably lead to increased chances for spatial overlap of successive fires, underscoring the need to understand how disturbances may interact and heightening ecological concern regarding the future of fire-prone forests (Stephens et al. 2013).

Multiple forest fires may interact as linked disturbances (Simard et al. 2011), such that the first fire can affect the likelihood of occurrence or severity (degree of ecological change caused by fire, typically measured by the amount of vegetation killed by fire [Keeley 2009]) of a second fire. By consuming fuels and killing plants that are less fire-resistant, a past fire may lower the probability that a second fire will ignite and/or spread (Teske et al. 2012, Parks et al. 2015), or if it does, that it will burn severely (Parks et al. 2014). In many areas, such negative interactions between forest fires can achieve goals of forest restoration and fuel treatments otherwise requiring widespread and persistent management intervention (North et al. 2015). In contrast, past fires that are followed by abundant growth of flammable vegetation (e.g., conifer seedlings or pyrophytic shrubs) may increase the

severity of a second fire (Thompson et al. 2007, Collins et al. 2009, Holden et al. 2010, van Wagtendonk et al. 2012). Such positive interactions can produce compound disturbance effects on ecosystem response (Paine et al. 1998) if a second severe (i.e., stand-replacing) fire occurs before sufficient recovery of propagules that are needed for post-fire recovery of dominant species (Brown and Johnstone 2012, Pinno et al. 2013). Extreme cases can drive transitions to alternative-stable-states whereby fire-prone vegetation begets more fire and/or erodes forest resilience (Odion et al. 2010, Larson et al. 2013).

Interactions between the severities of successive wild-fires have received attention in recent years, mostly consisting of localized studies using relatively few fire events or small areas (e.g., Thompson et al. 2007, Collins et al. 2009, Holden et al. 2010). Among existing studies, the detection of positive or negative interactions has varied, yielding findings that may be idiosyncratic to particular regions or events. Interactions between wildfires may differ based on the severity of the first fire, the interval between fires, and/or among different elevational forest zones. To test for consistent trends in these dynamics and to place findings in the context of the broader ecological understanding of other types of disturbance interactions requires a broad-scale regional analysis of fires across gradients of space and time.

In this study, we examine disturbance interactions between successive wildfires across the US Northern Rocky Mountains, a 300,000 km^2 region containing fire-prone forests representative of many northern-hemisphere temperate and boreal regions. We explore links between disturbances by testing the effect of past wildfire severity on the severity of subsequent wildfires, with additional attention to conditions that could lead to two stand-replacing fires occurring over a short interval. We focus on fire-severity outcomes given that a site burns twice, not the separate question of whether prior fires influence the likelihood of a second fire (see Parks et al. 2015). In locations that have experienced two fires between 1984 and 2010, we ask: (1) How do overall frequency distributions of burn-severity classes compare between first and second fires? (2) In a given location, how does burn severity of the second fire relate to that of the first? (3) Do interactions between successive fires vary by forest zone or the interval between fires? (4) What factors increase the probability of burning twice as stand-replacing fire? In general, we expected to find evidence of interactions such that frequency distributions in the second fires would be shifted toward lower severity classes, and severity at a given location would be lower in the second fire than the first fire. We expected this effect would attenuate at different rates and magnitudes depending on setting. Specifically, we expected greater magnitude and longer duration of negative interactions in low-elevation forests/woodlands with protracted post-fire fuel recovery. In contrast, in more productive mid- to high-elevation forests, we expected lower magnitude and shorter duration of negative interactions and greater potential for positive interactions with increasing intervals between fires.

METHODS

Study area

The US Northern Rockies ecoregion (EPA level III ecoregions 15, 16, 17, and 41) covers >30 million hectares (ha) in northwestern Wyoming, eastern Idaho, western Montana, and northeastern Washington (Fig. 1). Over 22 million ha (74%) of the US Northern Rockies is forest or woodland. At the highest elevations (~1,500–3,000 m), subalpine forests are dominated by subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*), lodgepole pine (*Pinus contorta* var. *latifolia*), and whitebark pine (*Pinus albicaulis*) accounting for 44.9% of the total forested area in the ecoregion (Appendix S1). At intermediate elevations (~750–2,500 m), mid-montane forests are dominated by Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), western larch (*Larix occidentalis*), ponderosa pine (*Pinus ponderosa*), limber pine (*Pinus flexilis*), and quaking aspen (*Populus tremuloides*) accounting for 52.7% of the total forested area in the ecoregion (Appendix S1). At the lower forest edge (elevations of ~500–1,500 m), low-montane woodlands contain sparse ponderosa pine, limber pine, and western juniper (*Juniperus occidentalis*) accounting for 2.5% of the total forested area in the ecoregion (Appendix S1). Fire regimes vary from high-frequency/low-severity fires at low elevations to low-frequency/high-severity fires at higher elevations; mid-elevations are characterized by mixed-severity fire regimes of variable frequency and severity (Habeck and Mutch 1973, Arno 1980, Romme and Despain 1989, Barrett et al. 1991, Barrett 1994, Kipfmueller and Baker 2000, Schoennagel et al. 2004).

Study design and data sources

Our sample universe included all overlapping burn perimeters (i.e., areas where at least two fire perimeters intersected) from a satellite-derived burn-severity atlas of US Northern Rockies forests (Harvey 2015), with data from the Monitoring Trends in Burn Severity (MTBS) Project (Eidenshink et al. 2007). This burn-severity atlas includes all fires that (1) occurred between 1984 and 2010 and had a perimeter within or intersecting the boundary of the Northern Rockies ecoregion, (2) were ≥50% forested, (3) were ≥250 ha in size, (4) were not a prescribed fire, and (5) occurred where neither pre- nor post-fire satellite images were obstructed by the scanner line correction failure on the Landsat 7 satellite. From burn-severity maps, we subsampled 30-m resolution pixels (hereafter "sample points") separated by ≥400 m to reduce potential for spatial autocorrelation; earlier work in the study region indicated that burn-severity pixels separated by ~400 m are spatially independent (Harvey et al. 2014*a*).
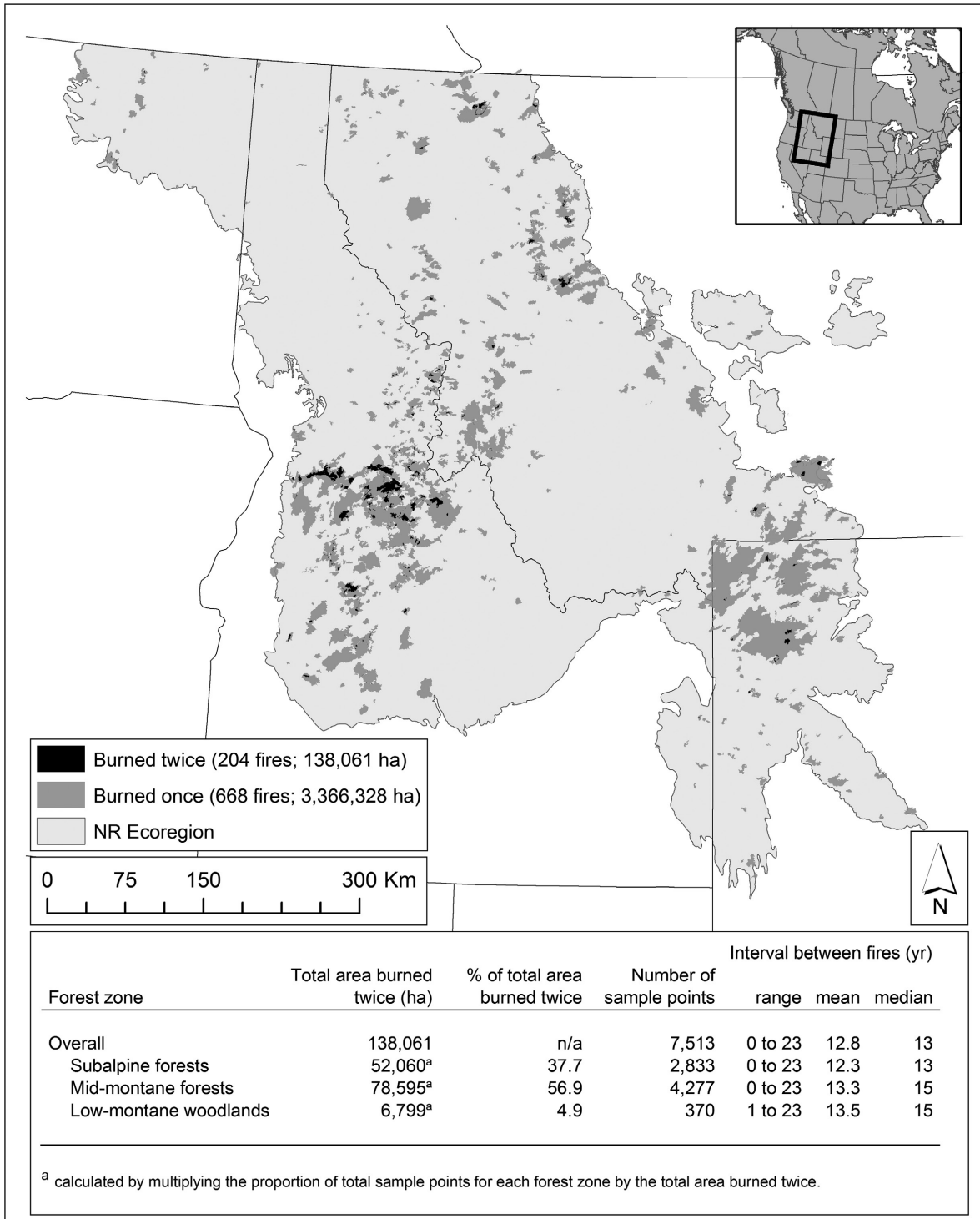
Forest zone	Total area burned twice (ha)	% of total area burned twice	Number of sample points	Interval between fires (yr)		
				range	mean	median
Overall	138,061	n/a	7,513	0 to 23	12.8	13
Subalpine forests	52,060[a]	37.7	2,833	0 to 23	12.3	13
Mid-montane forests	78,595[a]	56.9	4,277	0 to 23	13.3	15
Low-montane woodlands	6,799[a]	4.9	370	1 to 23	13.5	15

[a] calculated by multiplying the proportion of total sample points for each forest zone by the total area burned twice.

FIG. 1. Northern Rockies study area with the NR Ecoregion outlined in light gray, areas burned once in dark gray, and areas burned twice in black. Inset figure (upper right) shows regional location in North America. Inset table (bottom) contains data on total area (and by forest zone) burned twice in this study.

For each sample point, we extracted burn severity in the second fire as the focal response variable. Burn severity in all fires was measured using the Relative differenced Normalized Burn Ratio (RdNBR), a satellite index derived from Landsat imagery (Miller and Thode 2007). Because RdNBR accounts for differences in pre-fire vegetation biomass (including those associated with disturbances in the recent past), it is widely used to

detect relative changes caused by fire across extensive regions. The RdNBR index increases with greater burn severity, and validation of RdNBR in the US Northern Rockies has shown high correlations ($R^2 \sim 0.65$) with field measures of burn severity (Harvey 2015). Depending on our research question (see *Statistical analyses*), we used different characterizations of burn severity. Where continuous measures of burn severity were appropriate, we used RdNBR values. However, a limitation of RdNBR is that values have not been validated/calibrated to field measures of burn severity in many areas (Kolden et al. 2015). To overcome this limitation and for interpretation and relevance to forest/fire management, in most analyses we used field-validated burn-severity classes defined by thresholds of RdNBR associated with the percent of tree basal area killed by fire, measured in 371 field plots distributed throughout the study area (Harvey 2015). Categories were: low severity, <50% of basal area killed by fire, RdNBR ≤ 288; moderate severity, 50–92.5% of basal area killed by fire, RdNBR between 288 and 675; high severity (i.e., stand-replacing), >92.5% of basal area killed by fire, RdNBR ≥ 675 (Appendix S2). Breakpoints in RdNBR were similar to other regions (Cansler and McKenzie 2012).

Potential predictor variables were extracted for each sample point and represented prior fire activity, topographic setting, forest zone, and weather conditions during the second fire. For prior fire activity, we extracted burn severity (RdNBR or categorical, depending on research question) from the first fire that occurred within our study period, and the interval (yr) between that fire and the subsequent fire. For topographic setting, we extracted elevation (m), slope (degrees), and Topographic Moisture Index (cosine-transformed aspect as a proxy for moisture availability: 0 = dry, southwest facing; 2 = moist, northeast facing; [Beers et al. 1966]). For forest zone, we extracted Environmental Site Potential (ESP) from the LANDFIRE vegetation mapping project (Rollins 2009), which maps site potential for dominant vegetation cover by using consistent mapping techniques across all land ownership boundaries, and is not affected by previous disturbances. Each point was assigned a forest zone (subalpine forest, montane forest, low-montane woodland, Appendix S1) based on grouping of similar vegetation classes along an elevational gradient. For weather conditions during the second fire, we used the moisture deficit (mm) for the month(s) when the fire was burning, calculated from a 12 × 12 km gridded hydroclimatic dataset for the region (Westerling et al. 2011). Moisture deficit was normalized to departures from average conditions for each fire location by rescaling raw values to ±SD from mean values between 1981 and 2010 (Harvey et al. 2016). The spatial resolution of all variables was 30 × 30 m, except for weather variables, which were 12 × 12 km.

Prior to analysis, we excluded all sample points with RdNBR values <−500 or >2,000 in either fire; doing so excluded outliers but maintained >95% of sample points across the burn-severity spectrum. Sample points with RdNBR <−500 were unlikely to have experienced fire; rather they likely represent unburned patches within a burn perimeter. Sample points with RdNBR > 2,000 were likely an artifact of extreme reflectance values in the satellite (i.e., not related to burn severity). We also excluded sample points of any vegetation cover type other than our three focal forest zones (subalpine forests, mid-montane forests, low-montane woodlands). The resulting dataset contained 7,513 sample points.

Statistical analyses

For question 1 (distributions of burn severity in first vs. second fires), we used χ^2 tests of association/independence by combining all points in the first and second fires and testing for different frequencies among burn-severity classes. Separate tests were conducted to ask whether (1) the distribution of burn severity classes differed between the first vs. the second fires, and (2) the distribution of burn severity classes in the second fires was contingent upon burn severity class of the points in the first fire. For question 2, (interactions between burn severity in successive fires for a given location), we compared RdNBR values for the first and second fire at each sample point using paired t tests. Tests were conducted on all sample points combined, and separately for sample points within each forest zone. For question 3 (how burn-severity in successive fires varied by forest zone and fire interval), we used ordinal logistic regression where the response variable was the ordered burn-severity class in the second fire (low, moderate, high) and predictor variables were the burn-severity class in the first fire and fire interval (yr). Tests were conducted separately for each forest zone. Analyses for questions 1–3 included all sample points ($n = 7,513$).

In areas that burned as stand-replacing fire in the first fire ($n = 1,641$ points), for question 4 (factors that influence the likelihood of two stand-replacing fires), we used logistic regression with the response variable being the probability of burning as high severity (i.e., stand replacing) in the second fire. Predictor variables included elevation (m), slope (deg.), Topographic Moisture Index, burn severity of the first fire (RdNBR, over a narrower range, given high-severity in the first fire), interval between fires (yr), and moisture deficit during the second fire. To allow direct comparison of effect sizes among different variables in the model, each predictor and response variable was standardized to ±SD from the mean of that variable in the dataset.

Based on historical fire-return intervals (reviewed in Baker 2009), we assumed that first fires in our dataset occurred after longer intervals (>26 yr) than second fires (≤26 yr); potential implications are addressed herein. Analyses were conducted in R (R Development Core Team 2012). Ordinal logistic regression was conducted using the *'polr'* function in the MASS package (Venables and Ripley 2002). Results are means (±95% confidence intervals) unless noted.
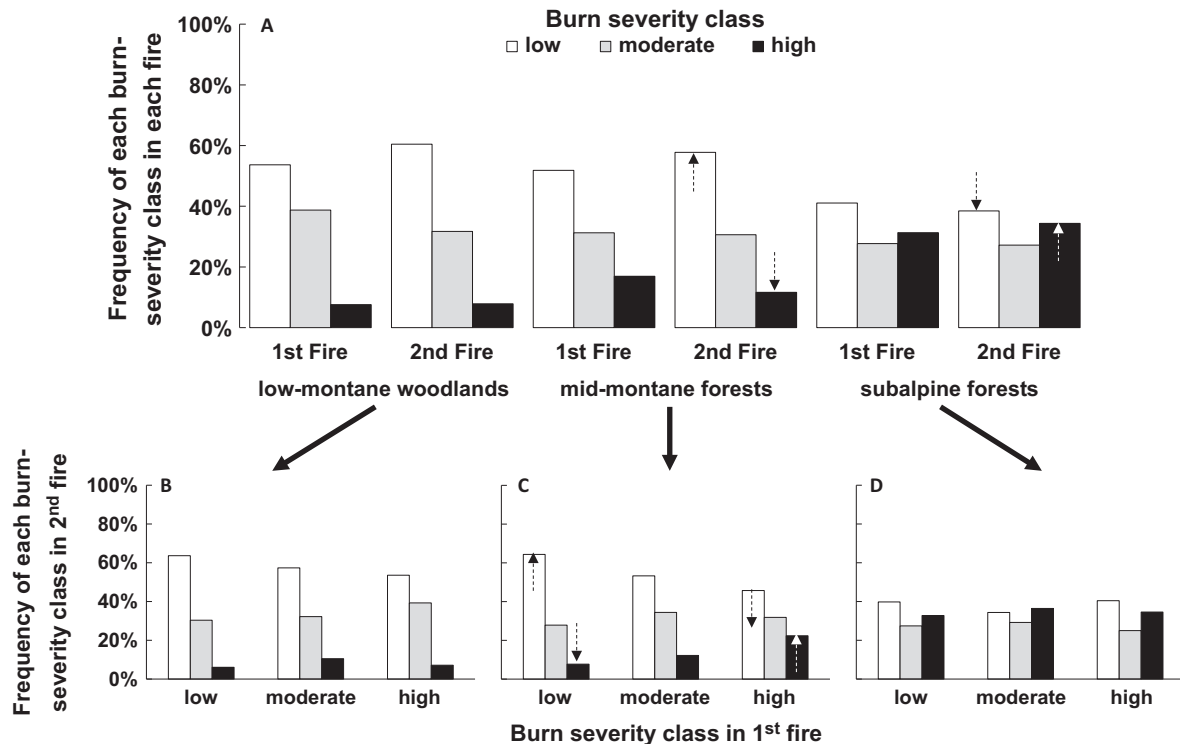
Fig. 2. (A) Frequency distribution of points that burned in each severity class for all points in each forest zone for the first fire and the second fire. Distributions differed by forest zone in the first fires ($\chi^2 = 256.5$, $P < 0.001$) and in all fires combined ($\chi^2 = 808.9$, $P < 0.001$). Within forest zones, distributions did not differ between the first and second fire for low-montane woodlands ($\chi^2 = 4.1$, $P = 0.13$), but did differ for both mid-montane ($\chi^2 = 55.9$, $P < 0.001$) and subalpine forests ($\chi^2 = 6.6$, $P = 0.04$). (B–D) Distribution of points that burned in each severity class in the second fire, contingent upon burn severity class of the first fire, separated by forest zone. The distribution of burn severity in the second fire was not associated with burn severity of the first fire for low-montane woodlands ($\chi^2 = 3.5$, $P = 0.48$), but was for mid-montane ($\chi^2 = 152.5$, $P < 0.001$) and subalpine forests ($\chi^2 = 9.5$, $P = 0.05$). Dashed arrows illustrate departure from expected frequencies.

Results

Overall, nearly 140,000 ha of forest in the US Northern Rockies burned two times between 1984 and 2010, with 204 distinct events (median and mean fire size = 81 ha and 677 ha, respectively; range = <1 to 13,089 ha; Fig. 1). The total area that burned twice was mostly in mid-montane forests (56.9% of total re-burn area), followed by subalpine forests (37.7% of total re-burn area) and low-montane woodlands (4.9% of total re-burn area). Mid-montane forests and low-montane woodlands constituted slightly more (and subalpine forests constituted slightly less) area of re-burn than expected based on their extent (Appendix S1). Intervals between successive fires ranged from 0 (two fires overlapping in the same year) to 23 yr, with a median interval of 13 yr across forest zones (Fig. 1; inset table). Among forest zones, the proportion of points in each burn-severity class differed for the first fire ($\chi^2 = 256.5$, $P < 0.001$) and for both fires combined ($\chi^2 = 808.9$, $P < 0.001$). In general, the proportion of low burn severity decreased and the proportion of high burn severity increased from low-montane woodlands to mid-montane forests to subalpine forests

(Fig. 2A); continuous burn-severity measures (RdNBR) correspondingly increased with forest-zone elevation (Fig. 3A–B).

Burn-severity class frequency distributions in successive fires

Overall frequencies of burn-severity classes in the first and second fires had similar distributions (Fig. 2), and although there were quantitative differences in distributions, effect sizes were small. Within forest zones, the distribution of burn-severity classes did not differ between first and second fires for low-montane woodlands ($\chi^2 = 4.1$, $P = 0.13$), but differed in mid-montane forests ($\chi^2 = 55.9$, $P < 0.001$) where high-severity decreased slightly in the second fire, and subalpine forests ($\chi^2 = 6.6$, $P = 0.04$) where high-severity increased slightly in the second fire (Fig. 2A). The distribution of burn severity in the second fire was not contingent upon burn severity of the first fire for low-montane woodlands ($\chi^2 = 3.5$, $P = 0.48$), but was for mid-montane ($\chi^2 = 152.5$, $P < 0.001$) and subalpine forests ($\chi^2 = 9.5$, $P = 0.05$; Fig. 2B–D). The most substantial difference was in mid-montane
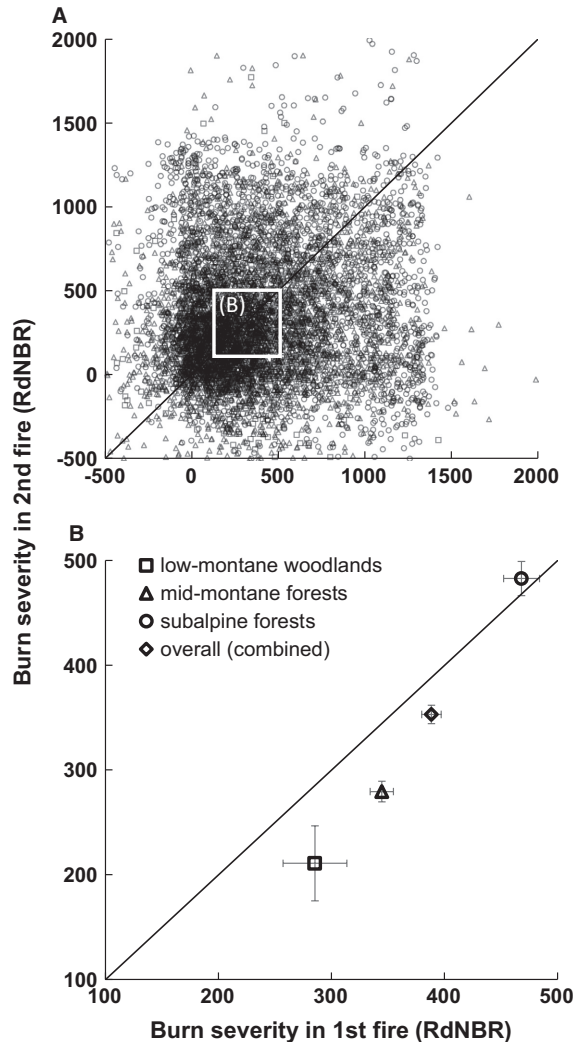
Fig. 3. (A) Relationship between continuous burn severity (RdNBR) in first fire and second fire that burned between 1984 and 2010 for each point sampled in this study. (B) Inset showing mean and 95% confidence intervals (error bars) for burn severity in the first (*x*-axis) and second (*y*-axis) fires. In both figures, diagonal 1:1 line represents equal burn severity in both fires. Below the diagonal line is lower burn severity in the second fire; above the line is higher burn severity in the second fire.

forests, where burning at low severity in the first fire was associated with greater probability of burning at low severity in the second fire, and burning at high severity in the first fire was associated with greater probability of burning at high severity in the second fire (Fig. 2C).

Burn-severity interactions in successive fires

Measured on a continuous scale (RdNBR), burn severity was highly variable within and among forest zones, and between the first and second fires at any given sample point (Fig. 3A). Across all forest zones, burn severity was 9 (±4) % lower in the second fire (353 ± 9) than in the first fire (388 ± 9; Fig. 3B, $P < 0.001$, paired

t test). The average reduction in burn severity between successive fires was strongest for low-montane woodlands (24 ± 20% lower; first fire: 285 ± 28; second fire: 211 ± 36; $P = 0.002$, paired *t* test, Fig. 3B), followed by mid-montane forests (19 ± 5% lower; first fire: 345 ± 10; second fire: 279 ± 10; $P < 0.001$, paired *t* test, Fig. 3B). For subalpine forests, there was no difference in burn severity between the first (468 ± 16) and second (483 ± 16) fire ($P = 0.22$, paired *t* test; Fig. 3B).

Interval-dependent effects of prior burn severity in each forest zone

For low-montane woodlands and mid-montane forests, low burn-severity in the first fire was associated with increased probability of low burn-severity (and corresponding decreased probability of high burn-severity) in the second fire; this effect did not vary with fire interval (Fig. 4A–B, left column). In contrast, high burn severity in the first fire led to an increase in the probability of low burn-severity in a second fire when fire intervals were <~10 yr, after which the probability of burning at moderate or high severity in a second fire became equal to or greater than the first fire (Fig. 4A–B, right column). For subalpine forests, low burn-severity in the first fire had no effect on the likelihood of any burn severity class in the second fire over all fire intervals (Fig. 4C, left column). High burn severity in the first fire led to a strong increase in the probability of low burn-severity in the second fire when fire intervals were <10 yr (Fig. 4C, right column). This effect was reversed when fire intervals exceeded ~10–12 yr, after which the probability of a second fire burning again at high severity increased sharply. Across all forest zones, effects of moderate burn severity in the first fire on the probability of burn severity class in the second fire were intermediate to those of low or high severity in the first fire (Fig. 4A–C, center column). Full model results are in Appendix S3.

Factors associated with successive stand-replacing fires

For areas that burned as high severity in the first fire, the probability of a second stand-replacing fire within our study period increased with elevation, topographic moisture index, interval between successive fires, and moisture deficit during the second fire; and decreased with slope and burn severity (RdNBR) of the first fire (Fig. 5; Appendix S4). The strongest effects were slope, interval between fires, elevation, and moisture deficit during the second fire, with effect sizes ~2 × stronger than topographic moisture index and burn severity of the first fire.

DISCUSSION

Links between burn severities in successive fires differ by forest zone and fire interval

Integrated over all fire intervals, the frequency distributions of burn-severity classes were remarkably similar
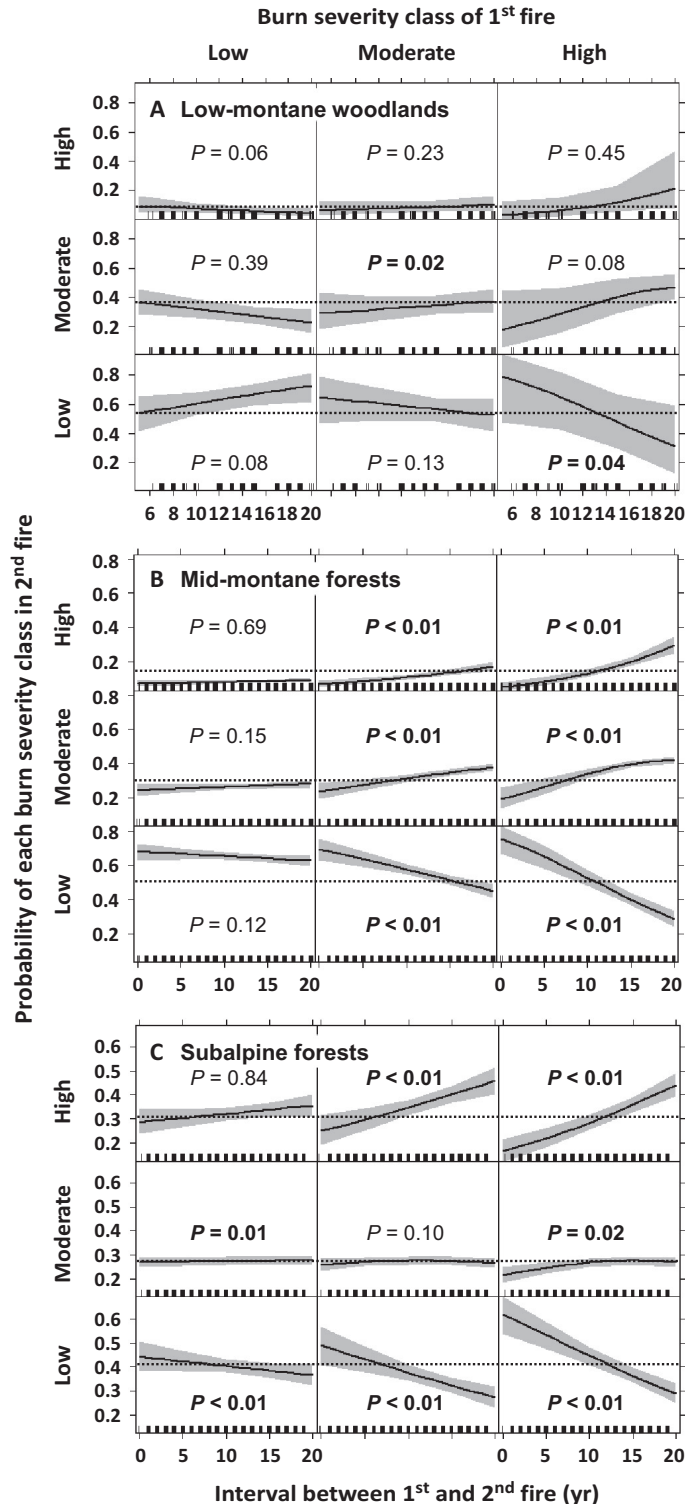
Fig. 4.   Effects of burn severity class in the first fire (columns from left to right) and time interval between fire events (*x*-axis within each column) on the probability (*y*-axis) of burning in each severity class in the second fire (rows from top to bottom), for (A) low-montane woodlands, (B) mid-montane forests, and (C), subalpine forests. Solid lines represent mean probability; gray-shaded envelopes represent 95% confidence intervals for probability. For comparison (i.e., control), dashed lines represent the probability of burning at each severity class (rows) in the first fire. Rugplots along each *x*-axis illustrate the number of datapoints at each interval between fires. *P*-values are for the slope of each line (with bold signifying *P* < 0.05), corresponding to the effect of time (interval) on the probability of burning in each severity class in a second fire. See Appendix S3 for model results.
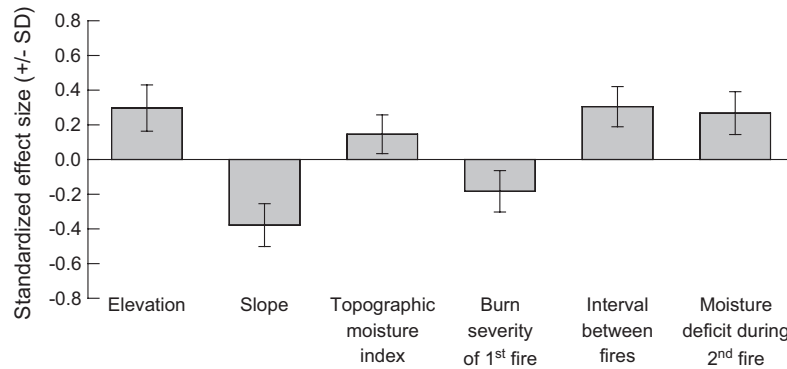
Fig. 5. Effects of each predictor variable on the probability of a location burning in a second stand-replacing fire, given that the first fire was stand-replacing and a second fire occurs. For comparison of importance among predictor variables, effects for each variable are standardized by centering on the mean and converting data points to ±SD from the mean. Bars represent mean effect size with error bars showing 95% confidence intervals. See Appendix S4 for model results.

between successive fires occurring within our study period, with a net effect of burn severity in the second fire ~9% lower than in the first fire. This relationship was stronger at low elevations and weaker at high elevations. Our findings suggest that forest fires that re-burn previously burned areas after short intervals are not qualitatively different in severity than fires burning at presumably longer intervals. However, our data show important differences in how interactions between fires vary depending on the severity of the first fire, the forest zone in which the fires occur, and the time interval between successive fires.

As expected, the effect of low-severity fire on the probability of subsequent high-severity fire differed by forest zone, suggesting important differences in forest and fuel structures between elevational zones and in the effect of prior fires on those structures. For subalpine forests, we found no effect of low-severity fire on subsequent fire severity, which contrasted with findings from the Southwestern US (Holden et al. 2010). Subalpine forests in the US Northern Rockies typically possess high tree density, large amounts of biomass, and very high fuel loads as such, fires are rarely fuel-limited (Schoennagel et al. 2004). Our results suggest that low-severity fire does not alter this fuel structure enough to affect future fire severity. In low-montane woodlands, we found a detectable, but relatively weak, effect of low-severity fire on subsequent burn severity. Low-montane woodlands possess sparse woody vegetation, low biomass, and low fuel loads therefore fires can be strongly fuel-limited (table 2.3 in Baker 2009). The small effect we detected suggests that there is not enough fuel to be substantially altered by low-severity fire. For mid-montane forests, however, we found a strong effect; low-severity fire greatly reduced the likelihood of subsequent high-severity fire over the duration of our study period. These results support findings from similar forests (Holden et al. 2010, van Wagtendonk et al. 2012, Parks et al. 2014, Kane et al. 2015), and suggest that low-severity fire has the strongest effect on subsequent fire severity when fuel loads are intermediate. Mid-montane forests have more woody

fuels than low-montane woodlands where warm/dry conditions and frequent fire support less woody biomass (table 2.3 in Baker 2009), but fewer woody fuels than subalpine forests where cool/moist conditions and infrequent fire support greater woody biomass.

Whether high-severity fire increased or decreased severity of subsequent fires depended on the interval between successive fires and varied with forest zone. Effects were strongest in mid-montane and subalpine forests, where past high-severity fire produced a strong, short-term (<10–12 yr) decreased likelihood of subsequent high-severity fire, followed by an equally strong increased likelihood of high-severity fire when fire intervals exceeded 10–12 yr. Similar findings have been reported from the US Southwest (Holden et al. 2010), Pacific Northwest (Thompson et al. 2007), and Sierra Nevada Mountains (van Wagtendonk et al. 2012). The lack of live biomass and combustion of most fine woody fuels following high-severity fire (Donato et al. 2013) is likely responsible for the immediate decrease in the probability of a subsequent fire burning at high severity within 10–12 yr of the first fire. Abundant post-fire tree regeneration produces rapid accumulation of live woody biomass within 10–20 yr of high-severity fire, particularly in productive sites (Donato et al. 2013, Turner et al. 2016). In less productive (i.e., drier) sites, rapid post-fire establishment of shrubs and grasses (native and non-native) can also produce highly flammable post-fire surface fuel loads (Merrill et al. 1980, Armour et al. 1984). Collectively, these dynamics in post-fire fuels likely govern the probability of burning with high severity in a second fire. Trends in low-montane woodlands were similar, but with wider confidence intervals and lower significance of the effect of fire interval.

Within each forest zone, effects of moderate burn-severity in the first fire on burn severity in a second fire were similar in direction to those for high-severity in the first fire, but consistently smaller in effect size. This is likely a result of less woody biomass killed (therefore contributing to lower post-fire dead woody fuel) and less

woody plant regeneration stimulated (therefore contributing to post-fire live woody fuels) following moderate-vs. high-severity fire.

When one disturbance is severe enough to remove key biotic components necessary for a second severe disturbance, linked interactions are typically strong and negative, with the duration of effects dependent on the biotic response following the first disturbance. Forests are less susceptible to severe fire shortly (<10 yr) following prior severe fire (this study, van Wagtendonk et al. 2012, Parks et al. 2014), as the key biotic components of live and dead fine fuels are removed by the first disturbance. The duration of this negative interaction is governed by the time needed for recovery of fuels. In fuel-limited lower elevations, negative interactions were stronger and longer-lasting (this study, Parks et al. 2014); in fuel-abundant higher elevations where fuels can recover quickly and are less-limiting to burn severity, negative interactions were short-lived and followed by strong positive interactions. Similarly, forests are less susceptible to severe beetle outbreaks for ~60 yr following prior severe beetle outbreaks (Hart et al. 2015) or ~100 yr following severe fire (Kulakowski et al. 2012). In both cases, the first disturbance removed the key biotic component necessary for a second severe disturbance (live, susceptible, host trees). When one disturbance does not remove the key biotic component(s) necessary for a second severe disturbance, linked interactions are relatively weak. Our finding in subalpine forests is a prime example, as low-severity fire did not affect subsequent burn severity (presumably because it did not significantly alter fuels) in this high-fuel-load system. Similarly, bark beetle outbreaks (even if severe) have little effect on the severity of subsequent fires (Harvey et al. 2013, 2014b), likely because partial disturbances (i.e., those that leave behind live non-host and non-susceptible trees) do not remove the key biotic component (fuels) for fire.

Satellite-derived measures of burn severity have two key methodological limitations. First, our dataset does not capture fires prior to 1984 (when Landsat TM satellite launched). As such, we cannot calculate fire intervals between the first fires in our study period and any previous fires. Given that 4% of all wildfires during our 26-yr study period were re-burns, however, our assumption that our first fires had pre-fire intervals longer than 26 yr should therefore be incorrect 4% of the time if burn rates remained constant over time. Much lower fire activity for the pre-1984 decades (compared to 1984–2010 [Morgan et al. 2014]) suggests that this error is <4%. Second, we know of no field validations of RdNBR (such as those we use, from mature forests sampled in Harvey [2015]) in burned areas of early post-fire vegetation. Because RdNBR is designed to detect relative burn severity across vegetation types (e.g., mature forest, shrubs, or young forest), it is likely better suited for re-burn analyses than non-relativized indices (e.g., dNBR), which are more affected by total pre-fire biomass (Miller and Thode 2007). With dNBR, Parks et al. (2014) found persistent (≤22 yr) effects of negative interactions between successive fires. However, live biomass is inherently lower following the first fire, and negative interactions may result from lower raw amounts of possible change in a second fire (Miller and Thode 2007).

Burn me twice, shame on who? Implications for resilience and management of fire-prone forests

The probability of burning twice at high severity was primarily driven by factors that contribute to rapid recovery of live biomass from a single stand-replacing fire: specifically, greater moisture availability associated with higher elevations and northeast facing aspects (Harvey et al. 2016). We also detected important effects of variation in severity in the first stand-replacing fire that likely affect post-fire recovery from the first fire. Stand-replacing crown fires can consume propagules in the canopy (e.g., in serotinous cones), reducing on-site seed sources for immediate post-fire tree establishment (Alexander and Cruz 2012). Surface fires, even if severe enough to be stand-replacing, can leave abundant seed sources behind (Turner et al. 1999, Larson and Franklin 2005, Harvey et al. 2014a), leading to dense post-fire vegetation and more fuels that may increase the likelihood of a second stand-replacing fire.

Weather conditions at the time of burning also mediate disturbance interactions. Under moderate weather conditions, prior fires can effectively limit the occurrence, spread, and size of subsequent fires, particularly if fire intervals are short (Collins et al. 2009, Teske et al. 2012, Parks et al. 2015). Such effects likely explain why re-burns accounted for only ~4% of the >3 million ha of forest that burned during our study period. Our finding that extreme weather (e.g., hot and dry conditions) can increase the likelihood of burning severely in the second fire has also been found with beetle outbreak–wildfire interactions (Harvey et al. 2014a). Weather conditions can override effects of fuels in conifer forests (e.g., Turner and Romme 1994, Bessie and Johnson 1995), and our results are consistent with such effects. If increased climate warming expands the frequency and duration of extreme fire-weather (Jolly et al. 2015), the extent of re-burns may increase as fuel-constraints become less important.

Positive feedbacks are an oft-posited agent of ecosystem decline, loss of resilience, or shifts to alternative states (Chapin et al. 1996). In our study, burn severity in the second fire was the dependent variable, and we did not measure ecosystem response. Thus, we cannot assess whether disturbance interactions might be contributing to directional state changes or altered feedbacks (e.g., Larson et al. 2013). The strongest positive interactions of burn severity we detected were in high-elevation sub-alpine forests, where high-severity fire is normal. If a second stand-replacing fire occurs before trees regenerating from the first fire have produced sufficient seed, tree regeneration may fail locally (Brown and Johnstone 2012, Pinno et al. 2013), with unknown consequences for long-term ecosystem feedbacks.

Our findings have important management implications when considering disturbance interactions in fire-prone forests under a warming climate. The data strongly support the assertion that low-severity wildfires in mid-montane forests and low-montane woodlands are akin to a "fuel treatment," or surrogate for prescribed burning (Stephens et al. 2009, Prichard and Kennedy 2014), in that they reduce the probability of subsequent severe fire if fire occurs within ~20 yr. Reducing risk of high-severity fire has long been a priority in such forests (Stephens et al. 2013), and these natural treatments can aid resource-limited forest management agencies struggling to accomplish fuels and restoration treatments across expansive areas (North et al. 2015). Conversely, low-severity wildfires do not alter burn severity of subsequent wildfires in subalpine forests, where fires are normally large, stand-replacing, and driven by weather and climate (Schoennagel et al. 2004). Therefore, using low-severity wildfire (or prescribed fires that produce similar effects; i.e., killing <50% of tree basal area) to lower the likelihood of subsequent severe fire in subalpine forests is not likely to be effective. Across all forest zones, high-severity fire can lower the likelihood of subsequent high-severity fire for ~10–12 yr, after which the potential for high-severity fire is equal to or greater than that of the first fire. High capacity for live-biomass recovery after one severe fire (e.g., abundant postfire tree regeneration on productive sites) thus paradoxically can lead to greater likelihood of a second high-severity fire. Conversely, areas that are slower to recover live biomass following the first severe fire are less likely to experience a second severe fire soon thereafter. As such, severe fires followed by sparse post-fire tree recruitment may be buffers against future severe fires.

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SUPPORTING INFORMATION